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Modelling the Hydrological Impacts of Climate Change on UK Lowland Wet Grassland

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Modelling the Hydrological Impacts of Climate Change on UK Lowland Wet Grassland

Abstract

Hydrological impacts of climate change upon the Elmley Marshes, southeast England, are simulated using a coupled hydrological/hydraulic model developed using MIKE SHE/MIKE 11 and calibrated to contemporary conditions. Predicted changes in precipitation, temperature, radiation and wind speed from the UK Climate Impacts Programme associated with four emissions scenarios for the 2050s are used to modify precipitation and potential evapotranspiration data. For each emissions scenario two sets of potential evapotranspiration data are derived, one using changes in temperature (PET_{temp}), the other incorporating changes in temperature, radiation and wind speed (PET_{trws}). Results indicate drier conditions through the progressively higher emissions scenarios when compared to contemporary conditions. Changes are particularly pronounced when using PET_{trws} . Summer water tables are lower (PET_{temp} 0.01–0.08 m; PET_{trws} 0.07–0.27 m) and the duration of high winter water tables is reduced. Although water tables still intercept the surface in winter when using PET_{temp} , this ceases when PET_{trws} is employed. Summer ditch water levels for the PET_{temp} scenarios are lower (0.01–0.21 m) and in dry winters they do not reach mean field level. Under the PET_{trws} scenarios summer and winter ditch water levels are lower by on average 0.21 m and 0.30 m respectively. Levels never reach the elevation of the marsh surface. Lower groundwater and ditch water levels result in declines in the magnitude and duration of surface inundation which is virtually eliminated with the PET_{trws} scenarios. Hydrological changes can be expected to have ecological impacts which may include the loss of some grassland species adapted to periods of high water table. Reductions in the extent of surface water in spring, especially for the PET_{trws} scenarios, are likely to reduce suitability for wading birds including lapwing (*Vanellus vanellus*) and redshank (*Tringa totanus*) for which the marshes are internationally renowned.

Keywords: climate change; hydrological/hydraulic modelling; wet grassland; wetlands

Introduction

Climate change will exert a strong influence upon wetlands throughout the 21st Century (e.g. Ramsar Bureau, 2002a, b). For freshwater wetlands the most pronounced impacts will be associated with modifications to hydrological regimes. These impacts will result from changes in the amount, state and seasonal distribution of precipitation, higher evaporation due to warmer temperatures and the combined impact of these changes upon runoff and groundwater levels (e.g. Hartig *et al.*, 1997; Mortsch, 1998; Conly and van der Kamp, 2001). Many freshwater wetlands are particularly vulnerable due to the delicate balance between precipitation and evaporation (Clair, 1998). For example, Dawson *et al.* (2001, 2003) suggested that in southern and central England and Wales raised and blanket bogs as well as wet heaths could be expected to be adversely affected by declining water availability. Particularly sensitive wetlands are those which are largely dependant upon precipitation and are isolated from other water sources such as inundation from streams and rivers, local runoff from upland areas or groundwater discharge. For example, lowland wetlands impounded by embankments, such as the site which features in this study, rely almost entirely upon precipitation (Hollis *et al.*, 1993). Changes in the amount of precipitation and its distribution through the year, coupled with enhanced evaporation, the largest flux of water from such wetlands, is likely to have direct consequences upon their hydrological regimes. These hydrological changes are likely to have knock-on ecological implications. For example, water level regime is an important control upon wetland plant communities (Wheeler *et al.*, 2004). Changes in groundwater depth, surface water levels and flood extent may therefore lead to shifts in vegetation which are adapted to particular hydrological conditions (Mortsch, 1998). Similarly hydrological conditions influence the use of wetlands by animals (e.g. Weller, 1994; Newbold and Mountford, 1997; van der Valk, 2006). Changes in water availability due to climate change may therefore impact the ability of some wetlands to support present animal populations which may be of conservation significance (e.g. Sorenson *et al.*, 1998; Herron *et al.*, 2002).

The Ramsar Bureau (2002c) has recognised the need to improve our knowledge of the vulnerability of wetlands to climate change. This paper addresses this need through the use of a coupled hydrological/hydraulic model to investigate the hydrological impacts of climate change upon the Elmley Marshes, a lowland wet grassland in southeast England. Predicted changes in meteorological conditions from climate modelling of different emissions scenarios are employed to modify model input data, enabling a comparison of contemporary and future hydrological regimes. Knowledge of the influence of hydrological

conditions upon wetland plants and animals, in this case birds, enables some inferences to be made regarding the potential ecological implications of climate change.

Methods

Study site: the Elmley Marshes

The Elmley Marshes are on the southern side of the Isle of Sheppey at the end of the Thames Estuary, southeast England (Figure 1), and are part of the wider North Kent Marshes, the largest tract of coastal wet grassland remaining in England and Wales (ADAS, 1997). They are typical of lowland wet grassland, a wetland type which includes semi-natural floodplain grasslands, grazing marshes, flood meadows, man-made washlands and water meadows (Jefferson and Grice, 1998; Joyce and Wade, 1998). Wet grasslands are characterised by periodic, but not continuous inundation, and permanently high water tables. They are predominantly located in river valleys, areas of impeded drainage or behind sea defences. The North Kent Marshes were created by the progressive enclosure and drainage of salt marshes. Evidence of these activities remains in the form of embankments which delineate the boundaries of earlier enclosures. Old embankments define the northern and eastern boundaries of the Elmley Marshes, whilst to the south and west they are bounded by the current sea defences. The embankments therefore effectively impound a discrete hydrological unit of approximately 8.7 km². Two small hills in the southeast rise to around 12 m above the marsh surface which itself has a mean elevation of 1.90 m above Ordnance Datum (m OD). There is a gradual decline in elevation towards the south where mean elevation is 1.75 m OD (Figure 2). Soils are of the Wallasea–Downholland association (Soil Survey of England and Wales classification 813f) and comprise pelo-alluvial gley soils derived from non-calcareous, clayey marine alluvium (Fordham and Green, 1980). High clay content results in low permeability and slow rates of water movement (Hazelden *et al.*, 1986).

The drainage system comprises a ditch network dividing the marshes into fields (Figure 2). It reflects the original salt marsh drainage, although ditches have been straightened, widened and deepened to improve drainage efficiency. Five main ditches cross the site into which secondary ditches converge. Additional drainage features are shallow linear features (rills) superimposed on field surfaces. These are remnants of small-scale salt marsh drainage channels. Tidal outfalls at the downstream ends of the main ditches discharge water at low tide into the Swale, a tidal channel separating the Isle of Sheppey from the mainland. Immediately upstream of these outfalls and at four other locations water control structures are installed within the ditches (Figure 2). These are “drop board sluices” comprising a grooved concrete spillway into which wooden boards are inserted or removed to control water levels (e.g. RSPB *et al.*, 1997).

Impoundment of the Elmley Marshes within embankments, coupled with low hydraulic conductivity, restricts surface and groundwater inflow so that seasonal differences in the relative importance of precipitation and evapotranspiration drive the hydrological regime. Macropores formed by summer desiccation of clay soils promote rapid autumn rise in water tables (Thompson *et al.*, 2004). Low-lying areas, including rills, are initially saturated and produce the first runoff to the ditches. Ditch water levels rise in response to this runoff and direct precipitation until water spills over the drop board sluices. At this time rills may become connected to ditches and act as pathways for water movement towards field centres (Thompson *et al.*, 2004). The water table is at, or close to, the surface for much of the winter and early spring creating a mosaic of dry land and shallow flooding which is supplemented by ponding of precipitation and runoff. Through spring and summer the water table and ditch water levels decline. Rills and other microtopographic depressions are the last areas to hold surface water. By mid-summer the water table reaches 0.80–0.95 m below the surface. Ditch water levels experience a similar decline and the shallowest may dry out.

Shallow water tables and winter flooding limited the traditional use of wet grasslands to extensive or low intensity agricultural activities associated with grazing and hay cutting. Grazing of the Elmley Marshes by sheep and cattle produces short- to medium-length grass swards. ADAS (1997) classed the dominant vegetation communities according to the National Vegetation Communities system (NVC, Rodwell, 1992) as MG6 (*Lolium perenne*/*Cynosurus cristatus* grassland), MG7 (*Lolium perenne* leys) and MG11 (*Festuca rubra*/*Agrostis stolonifera*/*Potentilla anserina* grassland). Specialist grassland species indicative of wet conditions within coastal grazing marshes include sea arrow grass (*Triglochin maritima*), divided sedge (*Carex divisa*) and saltmarsh rush (*Juncus gerardi*). Ditch flora reflects reductions in salinity inland from the tidal outfalls (Hollis *et al.*, 1993; Hollis and Thompson, 1998). Brackish communities with species including

sea club-rush (*Scirpus maritimus*), fennel pondweed (*Potamogeton pectinatus*), and soft hornwort (*Ceratophyllum submersum*) are found in most ditches, whilst fresher areas inland have more diverse flora. Species here include lesser reed-mace (*Typha angustifolia*), glaucous bulrush (*Schoenoplectus tabernaemontani*) and fool's water cress (*Apium nodiflorum*). Ditches also support a good invertebrate community. Beetles, dragonflies and damselflies are particularly well represented (English Nature, undated).

Low-intensity management coupled with high water tables and flooding have contributed to the importance of wet grasslands as wildlife habitat especially for waterfowl and wading birds (Ausden *et al.*, 2001; Ausden and Hirons, 2002). The North Kent Marshes support nationally important breeding populations of lapwing (*Vanellus vanellus*) and redshank (*Tringa totanus*). They are listed as internationally important under the Ramsar Convention, are included within a Special Protection Area under the EC Directive on the Conservation of Wild Birds (79/409/EEC) and include a number of Sites of Special Scientific Interest. The Elmley Marshes are a National Nature Reserve and are managed in line with agri-environmental schemes designed to promote ecological friendly farming including the maintenance of ecologically driven water levels and restoration of wet grassland in areas converted to arable. These measures aim to redress the loss of wet grassland, largely due to agricultural intensification and associated drainage and flood defence works, which took place during the second half of the 20th Century (e.g. Williams *et al.*, 1983; Williams and Hall, 1987; Mountford, 1994). These losses are implicated in the decline in wetland-related species including lapwing and redshank (Green and Robins, 1993; Ausden *et al.*, 2001).

Coupled hydrological/hydraulic modelling: current conditions

MIKE SHE is a deterministic, fully distributed and physically based modelling system (Refsgaard and Storm, 1995; Graham and Butts, 2005). It uses a finite difference approach to solve the partial differential equations describing overland (two-dimensional Saint-Venant equation), unsaturated (one-dimensional Richards' equation) and saturated subsurface flows (three-dimensional Boussinesq equation). Analytical solutions are used for describing interception, evapotranspiration and snow melt. Channel flow is simulated using MIKE 11, a one-dimensional hydraulic modelling system which can represent hydraulic structures including weirs, gates and culverts (Havnø *et al.*, 1995). The dynamic coupling of MIKE SHE and MIKE 11 evaluates for each time step river-aquifer exchange, overland flow to channels and flooding from channels to adjacent grid squares.

Thompson *et al.* (2004) provided a detailed account of the development, calibration, validation and results of a MIKE SHE/MIKE 11 model of the Elmley Marshes, one of the first applications of the coupled modelling system to a wetland. Data employed within the model are summarised in Table 1. The model area was divided into 9271 grid-squares of 30 m × 30 m with the elevation of each provided by a 1:2,500 topographic map (Figure 2). A single uniform saturated zone layer was specified and its hydraulic conductivity varied during calibration from an initial value guided by Al-Khudhairy *et al.* (1999) and Gavin (2001). A zero flow boundary around the model area was specified due to the impoundment of the marshes within embankments and the low hydraulic conductivity. The drainage option was used to represent runoff from topographic features too small to be shown in the model grid. A uniform soil profile with hydraulic properties based on Al-Khudhairy *et al.* (1999) was specified and included bypass flow to represent macropores. Precipitation and Penman-Monteith potential evapotranspiration (Monteith, 1965) were provided by an automatic weather station. These data were supplemented by a nearby rain gauge and the UK Meteorological Office Rainfall and Evaporation Calculation System (MORECS, Meteorological Office, 1992). Evapotranspiration parameters for a uniform grass cover were taken from the literature (Table 1). Overland flow resistance was a calibration term with initial values taken from Al-Khudhairy *et al.* (1999). The MIKE 11 ditch network was abstracted from 1:2,500 digital map data (Ordnance Survey Landline Plus). Cross-sections were based on field surveys, aerial photography and literature (Newbold *et al.*, 1989). Uniform channel roughness and leakage coefficients were used and were both modified during calibration. Rectangular weirs, with dimensions from LMIDB (1999), represented drop board sluices. The specification of positive flow only valves on the weirs at the downstream ends of the main ditches ensured they operated as tidal outfalls. Evaporation from ditches was represented as boundary conditions at the end of reaches which abstracted volumes of water derived from the product of daily evaporation rate and ditch water surface area. The latter were evaluated from water level/surface area relationships and water level records from stage boards and an automatic water level recorder (AWLR). The MIKE 11 model, including the location of cross sections, weirs and evaporation boundary conditions, is shown in Figure 2.

A 36 month simulation period (25/06/1997–29/06/2000) was divided into two 18 month sections for split sample calibration and validation (e.g. Klemeš, 1986; Refsgaard, 1997). This was based upon graphical comparisons of observed and simulated water table depths (obtained from piezometers) and ditch water levels from stage boards and the AWLR. Nash-Sutcliffe efficiency coefficients (R^2 , Nash and Sutcliffe, 1970) were also evaluated. Good agreement was obtained between model results and observed hydrological conditions. For example, Figures 4 and 6 provide representative observed and calibrated groundwater depths and ditch water levels (in addition to the result of climate change scenarios discussed below). The R^2 values for these comparisons are 0.80 and 0.83 respectively.

Coupled hydrological/hydraulic modelling: future climate scenarios

This study employed the climate impact assessment methodology advocated by Parry and Carter (1998) which has been widely used to assess hydrological impacts on river discharge (e.g. Chiew *et al.*, 1995; Viney and Sivapalan, 1996; Limbrick *et al.*, 2000; Menzel and Bürger, 2002). Arnell and Reynard (1996) outlined the stages that this approach involves:

- i. Define, calibrate and validate a hydrological model using current climate data;
- ii. Define climate change scenarios and perturb the original model input data accordingly;
- iii. Run the hydrological model using new input data and compare results with those obtained for current climate conditions.

The development, calibration and validation of the hydrological/hydraulic model of the Elmley Marshes by Thompson *et al.* (2004) satisfy the first of these stages. Results of this model provide the baseline conditions against which the impacts of climate change can be compared.

Climate change scenarios

This study uses climate changes predicted for the 2050s by the UK Climate Impacts Programme (UKCIP02, Hulme *et al.*, 2002). They are based on mean climate during a time slice covering 2041–2070 driven by four greenhouse gas emission scenarios described by the IPCC Special Report on Emissions Scenarios (SRES) (IPCC, 2000). Changes in climate parameters for the Low Emissions, Medium-Low Emissions, Medium-High Emissions and High Emissions scenarios (referred to here as L, ML, MH and H respectively) are referenced to a 1961–1990 baseline period. These changes are derived from a nested climate modelling approach in which a global climate model (HadCM3, resolution 250–300 km) provides boundary conditions for a global atmospheric model (HadAM3H, ~120 km) which in turn provides boundary conditions for a regional model of the European atmosphere (HadRM3, ~50 km). This dynamic downscaling provides a more appropriate resolution of climatic outputs from global climate models for use in hydrological impact studies (e.g. Kay *et al.*, 2006; Fowler and Kilsby, 2007).

Figure 3 summarises predicted changes in four climate parameters, expressed as average departures from the baseline period, for the Elmley Marshes for each emissions scenario for the 2050s time slice. The southeast of the UK, already the driest part of the country, is subject to some of the largest changes projected by the UKCIP02 scenarios. Figure 3a shows that summer (August) temperatures in the 2050s are projected to be 2.1°C warmer under the Low Emissions scenario and 3.3°C higher under the High Emissions scenario. Even in winter (January) temperatures are between 1.2°C and 1.9°C warmer. In winter precipitation increases whilst summers are drier (Figure 3b). These changes range from a 10.3%–16.4% increase in winter (January) to a 19.9%–31.6% decline in summer (July). Due to reduced cloud cover in every month except January downward shortwave flux, which comprises both direct and diffuse solar radiation, is higher (Figure 3c). This will further increase elevated evaporation rates resulting from higher temperatures. Changes in wind speed will also impact evaporation and Figure 3d shows that wind speed is predicted to increase by 2.8%–4.5% in winter and decrease by 1.9%–2.9% in summer. The degree of confidence in these projections is however lower than for some other parameters including temperature and precipitation (Hulme *et al.*, 2002).

Modification to model input data

Original daily precipitation data were multiplied by the monthly percentage changes provided by the UKCIP02 scenarios. Although this approach is relatively simple since it does not include changes in the

distribution and frequency of events (Chiew *et al.*, 1995), it has been widely adopted in hydrological studies of climate change (e.g. Arnell and Reynard, 1996; Arnell, 1999; Limbrick *et al.*, 2000; Kamga, 2001). Two new evapotranspiration time series were derived for each emissions scenario. Using the approach of Arnell and Reynard (1996), the first assumed only a change in temperature. Temperature data from the automatic weather station were modified in accordance with the monthly UKCIP02 temperature changes and daily evaporation recalculated using the Penman-Monteith formula. These data are referred to as PET_{temp} . The second evapotranspiration time series, referred to as PET_{trws} , used the same approach but incorporated changes in temperature, net radiation and wind speed. Both evapotranspiration times series, and hence the simulations of climate change, were limited to the period 25/06/1997–28/03/2000 due to the removal of the automatic weather station in late March 2000. Since the simulation period falls outside the baseline period against which changes are referenced by the UKCIP02 scenarios, modified input data are likely to be representative of conditions towards the latter part of the 2041–2070 time slice.

Table 2 provides total annual precipitation and potential evapotranspiration for the two complete hydrological years (September–August) of the simulation period for the original model data and each emissions scenario. Relatively modest declines in annual precipitation are evident, although these mask seasonal changes discussed above. In contrast, for all emissions scenarios, every month witnesses increased potential evapotranspiration contributing to enhanced annual totals especially for PET_{trws} . The annual net precipitation (precipitation - potential evapotranspiration) figures in Table 2 demonstrate the drier conditions from the observed meteorological data through the progressively higher emissions scenarios, particularly in the case of PET_{trws} . A dry year such as 1997/8 (long-term mean precipitation is 530 mm) is especially impacted. When evaluated on a monthly basis, changes in net precipitation are positive in only eight (PET_{temp}) or seven (PET_{trws}) months of the 33 month simulation period. The magnitudes of these increases, which are confined to months between November and February, are small in comparison to the decreases throughout the rest of the year.

Incorporation of modified input data within coupled hydrological/hydraulic model

For each emissions scenario two model runs were simulated using the modified precipitation and the two potential evapotranspiration time series (PET_{temp} and PET_{trws}). Using the Low Emissions scenario as an example, these runs are referred to as L_{temp} and L_{trws} respectively. The incorporation of modified precipitation and potential evapotranspiration within MIKE SHE simply required the specification of the relevant time series. However, the method employed to represent evaporation from the MIKE 11 ditches required the re-evaluation of the evaporation boundary conditions. Thompson (2004) showed that ditch evaporation was insensitive to ditch water level due to the steep sided MIKE 11 cross-sections. Therefore new evaporation boundary conditions were derived using the approach employed by Thompson *et al.* (2004) but replacing observed ditch water levels with the mean observed level of 1.07 m OD.

Results

Groundwater

Figure 4 shows simulated groundwater depths from each of the climate change runs grouped according to the modified potential evapotranspiration data used (PET_{temp} and PET_{trws}). It also shows results from the calibrated model for the same MIKE SHE grid square and observations from a piezometer installed at this location. The simplicity of the hydrogeological conditions within the model and low gradients mean that groundwater depths for each simulation are representative of those in the low-lying part of the model area.

Under all the PET_{temp} scenarios the water table still rises to intercept the surface in the first two winters of the simulation period despite falling further at the end of the preceding summers. However, this rise is delayed, especially in 1998/9. Table 3 summarises the number of days in each complete hydrological year when the simulated water table was above threshold depths close to the ground surface. In 1997/8 the calibrated water table was within 0.01 m of the surface for nearly 70 days and within 0.10 m for nearly 120 days. However, under all four PET_{temp} scenarios the water table falls in mid-February and, although rising in March, does not intercept the ground again until mid-April. The number of days when the water table is close to the surface therefore declines. Although there is a general reduction in the number of high groundwater days in 1997/8 from L_{temp} to H_{temp} , the results for MH_{temp} are an exception. Increased winter precipitation results in small

increases in net precipitation for H_{temp} compared to MH_{temp} despite higher evapotranspiration rates. This prolongs the period of high water table for the former scenario compared to MH_{temp} . In the winter of the following year, PET_{temp} water tables have to rise further still before intercepting the surface which they do between 40 (L_{temp}) and 56 (H_{temp}) days after the calibrated results. Under the H_{temp} scenario the spring water table falls 37 days earlier compared to the other scenarios. The duration of periods when water tables are close to the surface declines from the calibrated results through each of the progressively higher emissions scenarios. Declines associated with H_{temp} are particularly large. Water tables during the following spring and summer are again lower for the PET_{temp} scenarios whilst the gains in groundwater elevation in autumn/winter of 1999/2000 are very subdued compared to calibrated results.

Changes in groundwater depth are more pronounced for the PET_{trws} scenarios (Figure 4b). Differences between results of the four climate change scenarios are particularly large in winter. Higher spring and summer evapotranspiration causes even lower groundwater levels at the start of autumn and, despite increases in winter precipitation, the water table fails to recover to the elevations evident in calibrated results. The water table rarely approaches the ground surface. Only under the L_{trws} scenario does it reach within 0.10 m of the surface and then for only very short periods (Table 3). Under the H_{trws} scenario, the water table never comes within 0.20 m of the surface whilst it only does so under the ML_{trws} and MH_{trws} scenario in 1997/8. Only very small gains in groundwater level occur in the autumn/winter of 1999/2000 when calibrated results show rapid gains in water table elevation.

Table 4 shows minimum, maximum and range of groundwater depths for the calibrated model and each climate change scenario for both complete hydrological years of the simulation period. Since the recession in water tables extends beyond the end of the usual hydrological year (September–August), this analysis defines the hydrological year as the period from the lowest water table before one autumn/winter rise to the lowest water table the following year. Peak winter groundwater levels in all of the PET_{temp} scenarios in both years are the same since the water table intercepts the ground surface (Table 4 provides the depth of surface water when this occurs but since flooding results from groundwater intercepting the surface, ponding of precipitation and inundation from ditches the minimum groundwater depth is assumed to be 0.0 m). Consequently, lower summer water tables lead to small increases in the seasonal range of groundwater depths. In contrast, the seasonal range of groundwater depths declines from the Low through to the High emissions scenarios which use PET_{trws} . Groundwater changes are illustrated further by depth–duration curves for each scenario and the calibrated model derived using results from the complete simulation period (Figure 5). Drier conditions for the progressively higher emissions scenarios are demonstrated as is the more extreme drying trend for the PET_{trws} scenarios. Increases in the range of groundwater depths for the PET_{temp} scenarios are evident (Figure 5a) as is the virtual elimination of saturated conditions at the ground surface and the reduction in range of groundwater depths for the PET_{trws} scenarios (Figure 5b).

Ditch water levels

Figure 6 shows simulated ditch water levels for the eight climate change scenarios grouped according to the potential evapotranspiration data used (PET_{temp} and PET_{trws}). Results from the calibrated model for the same location as well as a stage board installed in this ditch are also shown. For each scenario the results are representative of those throughout the MIKE 11 model in which water levels are approximately uniform due to the low gradients and inter-connected nature of the ditch network.

Figure 6a shows that for the four PET_{temp} scenarios a general lowering of ditch water levels occurs throughout most of the simulation period. Spring and summer draw downs are larger for each of the progressively higher emissions scenarios. Despite increased precipitation between November and March, ditch water levels for all PET_{temp} scenarios fail to reach those experienced under calibrated conditions in the winter of 1997/8 and the initial rapid rise is delayed by around 10 days. Whereas the calibrated ditch water level exceeds the elevation of the MIKE 11 weirs (1.75 m OD) on 40 days in 1997/8, this does not occur once under any of the PET_{temp} scenarios. Although ditch water levels in 1998/9 exceed the weir elevation under all the PET_{temp} scenarios the rise is progressively delayed (first reaching 1.75 m OD 17 and 40 days after the calibrated results for L_{temp} and H_{temp} respectively). Some short-lived peaks resulting from individual rain events which increase in magnitude under climate change exceed calibrated ditch water levels. Water levels begin to recede earlier in the spring of 1999 so that the number of days when the elevation of the weirs is exceeded declines. Under calibrated conditions ditch water would overtop the weirs on 78 days in 1998/9.

The corresponding figures for L_{temp} and H_{temp} are 65 and 15 respectively. Towards the end of the simulation period, the rapid rise in ditch water levels shown in the calibrated results is delayed and reduced in magnitude under the progressively higher emissions scenarios and is absent in the H_{temp} results.

Differences between calibrated ditch water levels and those associated with climate change scenarios are larger when using PET_{trws} and the overall drying trend is enhanced considerably (Figure 6b). The magnitude of annual drawdown increases systematically from L_{trws} through to H_{trws} and the duration of the drawdown increases from year to year so that the autumn/winter gains in water level are delayed. These increases in water level are of similar magnitude to those of the calibrated results in 1997/8 whilst in the following year they exceed the rise in calibrated ditch water levels. However, the lower initial levels ensure that in both years ditch water fails to reach the elevation of the MIKE 11 weirs. Peak winter ditch water levels are very similar for each climate change scenario and each hydrological year. A rise in ditch water levels in the winter of 1999/2000 is absent from the results of all four PET_{trws} scenarios.

The ditch water level–duration graphs of Figure 7 summarise the changes resulting from the climate change scenarios. The overall lowering of water levels from the calibrated through the Low to the High emissions scenario for both the PET_{temp} and PET_{trws} simulations is demonstrated as are the larger reductions in ditch water levels associated with the PET_{trws} scenarios. In all cases the range in ditch water levels increases compared to calibrated results. For the PET_{temp} scenarios there is little difference in the range of ditch water levels between emission scenarios with the exception of H_{temp} which has a wider range due to the absence of any recovery in the winter of 1999/2000. In contrast, the overall ditch water level range increases progressively from L_{trws} to H_{trws} .

Surface inundation

Inundation of the marsh surface results from a combination of high water tables which intercept the ground surface especially within low-lying rills, movement of water from ditches often along these rills, and ponding of precipitation and local runoff (Thompson *et al.*, 2004). The climate change scenarios reduce the incidence of conditions conducive to surface flooding by lowering groundwater and ditch water levels and by reducing the duration of periods of high water level when they do occur. Figures 8 and 9 show the extent of open water for a range of maximum depths for the PET_{temp} and PET_{trws} scenarios respectively. The corresponding data for the calibrated results are also shown.

The extent and duration of shallow (maximum depths 0.1 m and 0.2 m) flooding under the PET_{temp} scenarios (Figures 8a&b) reflect changes in both groundwater and ditch water levels. The area of shallow flooding is reduced whilst the progressively longer delay in the expansion of flooding in 1998/9 results from delayed gains in groundwater and ditch water levels. Similarly, the earlier recession in water levels cause shallow inundation to recede earlier. Depleted groundwater levels between mid-February and mid-April 1998 reduce inundation at this time with the middle peak in flooding of 0.2 m maximum depth shown in the calibrated results being noticeably absent (Figure 8b). Some shallow inundation is retained through most of the spring and summer although its extent is reduced. Figure 8c shows that in 1997/8 the one distinctive peak in flooding of 0.3 m maximum depth shown in the calibrated results is absent from all the PET_{temp} scenarios. Thompson *et al.* (2004) suggested that a threshold ditch water level of 1.75 m OD, coupled with a high water table, is associated with the expansion of flooding of this depth. This threshold was not exceeded in 1997/8 for any of the PET_{temp} scenarios. In the following year the threshold was exceeded under all the PET_{temp} scenarios although some time after the calibrated results. Some small winter peaks exceeded those of the calibrated results. Figure 8c shows short-lived periods for three of the PET_{temp} scenarios (L_{temp} , ML_{temp} and MH_{temp}) when the extent of flooding of 0.3 m maximum depth exceeds that shown for the calibrated results although the maximum extent of flooding of this depth is still associated with the calibrated model. On three occasions in 1998/9 the extent of flooding of 0.4 m maximum depth under calibrated conditions is exceeded by some climate change scenarios (L_{temp} and/or ML_{temp}) (Figure 8d). However, these increases are small and the predominant trend is for progressively smaller areas of deeper flooding from L_{temp} to the H_{temp} . In 1997/8 the extent of areas flooded to a maximum depth of 0.4 m is generally consistent between PET_{temp} scenarios and is lower than under calibrated conditions, when these areas were already small and restricted to locations immediately adjacent to ditches in the far south of the marshes.

The much larger changes in groundwater and ditch water levels for the PET_{trws} scenarios lead to dramatic reductions in flood extent. Figure 9a shows that in 1997/8 there are three peaks in shallow (0.1 m) flooding of approximately the same magnitude but much reduced compared to the calibrated results. Not all of these peaks are present in results of the four scenarios. Both the L_{trws} and ML_{trws} scenarios yield the first, only L_{trws} provides the second whilst all four scenarios display the third peak. Thompson *et al.* (2004) identified a threshold ditch water level of 1.57 m OD required to initiate shallow inundation. High groundwater levels also promoted inundation although a distinct threshold water table depth was less discernable. Figure 6b shows three distinct peaks in ditch water levels for the PET_{trws} scenarios during the 1997/8 period of elevated water levels. Those which exceed the 1.57 m OD threshold for more than a day or two correspond with the peaks in shallow flooding (Figure 9a). In 1998/9 the extent of shallow flooding is small and consistent between the PET_{trws} scenarios. Its maximum extent is less than 10% of the corresponding area for calibrated results. Ditch water levels for most of the PET_{trws} scenarios failed to reach the 1.57 m OD threshold in 1998/9 and for the one scenario where they did (L_{trws}) levels only exceeded this threshold by a very narrow margin and for short periods. Figure 9a shows that in the spring and summer flooding of 0.1 m depth disappears completely. All of the PET_{trws} scenarios have eliminated significant areas flooded to a maximum depth of 0.2 m and 0.3 m (Figures 9b&c). Ditch water levels fail to reach the 1.65 m OD and 1.75 m OD thresholds identified by Thompson *et al.* (2004) which are required for flooding of these depths. Similarly, the extent of deeper (0.4 m) inundation is further restricted under the PET_{trws} scenarios (Figure 9d). The peak extent of inundation of this depth varies little between scenarios, implying that the same areas are flooded, although the duration of its presence tends to be longer for the lower emissions scenarios.

Discussion

Changes in the hydrological regime of the Elmley Marshes revealed by the simulation of climate change scenarios are summarised in Table 5. Climate change has the effect of tipping the balance between precipitation and evaporation in favour of evaporation thereby inducing drier conditions. The autumn/winter rise in water table and ditch water levels are delayed whilst in the spring levels begin to decline earlier. Periods of high water levels are therefore shorter. Consequently the extent and duration of extensive surface inundation both decline. Spring and summer recessions in groundwater and ditch water levels are enhanced and continue for longer into the following autumn. The magnitude of these changes generally increase through the progressively higher emissions scenarios and are particularly pronounced when predicted changes in temperature, net radiation and wind speed are used to modify evapotranspiration. In these cases winter groundwater levels rarely approach the ground surface limiting inundation. Flooding is further impacted for these PET_{trws} scenarios as ditch water levels do not reach the threshold levels required to induce movement of water from the ditches onto the marsh surface. Where enhanced winter precipitation associated with predicted climate change raises water levels, these changes are modest, especially when compared to the magnitude of reductions in water levels at other times, and short-lived. The impacts of climate change upon the hydrology of the Elmley Marshes are likely to have a number of ecological implications. Since it is not unreasonable to suggest that other parts of the North Kent Marshes and similar wetlands in southeast England can be expected to experience broadly similar changes in their water level regimes, these ecological impacts are of relevance beyond the Elmley Marshes.

Wetland plant communities and species have specific and critical ecohydrological requirements which include water level regime (Wheeler and Shaw, 1995; Wheeler *et al.*, 2004). Wheeler *et al.* (2004) advocated the combination of water level requirements of wetland plant communities or species and predicted hydrological changes from modelling to evaluate impacts of climate change or other scenarios involving, for example, abstraction or restoration. They suggested that this approach could establish whether vegetation is 'out of regime' or is at risk of moving out of regime in terms of water needs. This approach does, however, require hydrological models capable of accurately simulating water table elevations at a resolution which is commensurate with hydrological changes which would induce ecological impacts. Results of the present study provide such an opportunity for examining potential ecological changes of the climate change scenarios. The Sum Exceedence approach proposed by Sieben (1965) and adapted to wet grassland communities by Gowing *et al.* (1998) confirms that the soil water regime of the calibrated model, which represents current conditions, corresponds to floodplain grassland experiencing moderate stress from soil anoxia resulting from high water tables. In the Elmley soils anoxia is likely to occur when water tables are shallower than 0.3 m as air-filled porosity of the root zone falls below 10% (Taylor, 1949). All four PET_{temp} scenarios result in stress from anoxia being reduced by approximately half while the PET_{trws} scenarios

remove this stress entirely. Absence of soil anoxia will favour grassland communities that are typical of well-drained fertile soil (e.g. MG6) at the expense of those tolerant to waterlogging (e.g. MG11). The latter currently characterise the Elmley Marshes and include specialist species of coastal grazing marshes such as sea arrow grass (*Triglochin maritima*), divided sedge (*Carex divisa*) and saltmarsh rush (*Juncus gerardi*). These species, which rely on periods of soil anoxia, are likely to be lost from the sward and replaced by more generalist species typical of dry grassland. Ditch vegetation may also be impacted by climate change induced lower water levels. More ditches are likely to dry out in summer whilst an expansion of brackish conditions into areas which are currently relatively fresh and support more diverse flora may also occur.

Drier conditions are also likely to impact wader populations which provide the marshes with particular conservation significance. Distributions of wading birds on wet grasslands have been shown to be strongly related to surface wetness (e.g. Eglington *et al.*, 2008). The probability of a particular part of the marshes being occupied by redshank and lapwing during the breeding season (April–June), as well as wader density, increases with flood extent and the number of wet rills and hollows (Milsom *et al.*, 2000, 2002). Feeding rates are also higher in rills which are wet in May compared to those which are dry (Milsom *et al.*, 2002). This may result from the effects of prolonged inundation on vegetation cover, availability of aquatic invertebrates or more penetrable wet soil. Under calibrated conditions there is extensive shallow inundation in April and May (Figures 8 and 9). The mean extent of inundation of 0.1 m maximum depth during these months for 1998 and 1999 is 1.21 km² and 1.13 km² respectively (Figures 10a&b). Flooding at this time is concentrated in the southern low-lying parts of the marshes and further north alongside ditches. Isolated topographic lows are also flooded (Figure 10c). For the L_{temp} and H_{temp} scenarios the mean extent of shallow flooding in April and May declines by 25.8% and 48.1% respectively (Figure 10a). Relatively large areas in the south and adjacent to ditches are still inundated although away from these locations flood extent declines especially for H_{temp} (Figure 10d&e). Maintenance of some inundation suggests that, at least in terms of water requirements, the marshes would be able to support lapwing and redshank although numbers may decline. Much larger reductions in breeding season flooding result from the PET_{trws} scenarios (Figure 10b). The mean reduction compared to calibrated results for 1998 and 1999 are 79.5% and 90.8% respectively. Flooding at this time is restricted to small low-lying areas adjacent to ditches in the far south of the marshes (Figures 10f&g). The national press has highlighted the plight of lapwing, redshank and other waders due to drought at locations in southeast England including the Elmley Marshes (The Times, 2005; BBC, 2006). Lapwing and redshank numbers on the marshes plummeted in 2005 due to low spring water levels (Burstons, 2006). The PET_{trws} scenarios suggest a shift away from the hydrological conditions which have favoured wading birds and a decline in the ability of the marshes to support large numbers of lapwing and redshank. Since similar hydrological changes could be expected throughout the North Kent Marshes the wider populations could be compromised.

Opportunities for tackling the hydrological impacts of climate change within the Elmley Marshes are limited. Maintenance of relatively high surface water levels throughout spring and summer will rely on storing as much water as possible in winter to counteract higher evaporation rates. However, Thompson (2003, 2004) showed that current water level management, which sets the drop board sluices at mean field level, approaches the optimum in terms of maintaining high ditch water levels and inundation. Even under current climate conditions only very modest increases in ditch water level and flooding could be achieved by raising sluices. Mitigating the impacts upon groundwater levels will be more problematical. The low soil hydraulic conductivity limits interactions between ditches and shallow groundwater (Gavin, 2001; Thompson, 2004). Even if predicted declines in ditch water level could be reduced, there would be little impact on the lower water tables which would result from enhanced evapotranspiration. Flooding the marsh surface supplements the water table but the extent of inundation has been shown to decline progressively under each of the higher emissions scenarios whilst, as noted above, there are limited possibilities for enhancing inundation with existing infrastructure. Limiting discharge through tidal outfalls might offer a partial solution to raising ditch water levels, inducing inundation and supplementing the water table, although this would need careful management. Bunds or low earth embankments have been used in another wetland National Nature Reserve on the Isle of Sheppey (English Nature, 1991) and elsewhere in the UK (RSPB *et al.*, 1997) to retain and control water levels within relatively small areas. The costs of implementing this approach for larger areas may be prohibitively large. Similarly, the long-term sustainability of supplementing water levels by pumping from the underlying chalk, an approach adopted elsewhere on the Isle of Sheppey, is doubtful.

The marshes face other climate change impacts. Relative sea levels in southeast England are predicted to be between 19 cm and 79 cm higher by the 2080s compared to 1961–1990 (Hulme *et al.*, 2002). This is likely to limit the period when gravity drainage through tidal outfalls is possible. Silting up of outfalls from an adjacent marsh has already necessitated the installation of pumped drainage (Hollis *et al.*, 1993). The reduction in outflows might, at least initially, act to mitigate impacts of climate change on water levels by facilitating water storage. However, the ability to manage water levels would be limited, and ecologically damaging water levels and flood durations might result. Higher sea water levels will increase the chance of embankments being overtopped and will add to debates over whether sea defences should be maintained to protect what is, at least from an agricultural point of view, less economically valuable land (e.g. Ledoux *et al.*, 2005). The ultimate fate of the marshes may be the replacement of freshwater wetlands with saline ecosystems, a trend forecast for other coastal areas (e.g. Mulrennan and Woodroffe, 1998; Eliot *et al.*, 1999).

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Tables

Parameter	Sources
Precipitation	Elmley Marshes automatic weather station; Barnlands rain gauge.
Potential evapotranspiration	Elmley Marshes automatic weather station; MORECS (Meteorological Office, 1992).
Topography	1:2,500 topographic map derived from stereoscopic aerial photography.
Soil hydraulic characteristics	Field analyses (Hamm, 1998; Gavin, 2001); Al-Khudhairy <i>et al.</i> (1999).
Vegetation characteristics	Thompson <i>et al.</i> (1981); Bultot <i>et al.</i> (1990); Kelliher <i>et al.</i> (1993).
Groundwater depth	Piezometer observations.
Ditch network	1:2,500 Ordnance Survey Landline Plus digital data.
Ditch cross-sections	Field surveys; Aerial photography; Newbold <i>et al.</i> (1989).
Control structure dimensions	LMIDB (1999).
Ditch water levels	Stage board and automatic water level recorder observations.

Table 1. Data employed within the coupled MIKE SHE/MIKE 11 model of the Elmley Marshes

Hydrological year	Parameter	Obs [†]	Emissions scenario			
			L	ML	MH	H
1997/8	Precipitation	460.0	455.5	454.7	454.1	452.9
	PET _{temp}	600.0	632.7	638.7	643.3	651.9
	PET _{trws}		688.9	706.0	719.2	744.1
	Net Precipitation (PET _{temp})	-140.0	-177.1	-184.0	-189.2	-199.0
	Net Precipitation (PET _{trws})		-233.4	-251.3	-265.1	-291.2
1998/9	Precipitation	599.4	564.8	558.5	553.5	544.5
	PET _{temp}	608.4	640.9	646.9	651.5	660.0
	PET _{trws}		703.1	721.3	735.4	762.0
	Net Precipitation (PET _{temp})	-9.0	-76.1	-88.5	-98.0	-115.6
	Net Precipitation (PET _{trws})		-138.3	-162.9	-181.9	-217.5

[†] Observed data employed in the original model of the Elmley Marshes by Thompson *et al.* (2004).

Table 2. Annual precipitation, potential evapotranspiration and net precipitation for the Elmley Marshes in 1997/8 and 1998/9: Observed and modified according to UKCIP02 climate change projections for 2050

Hydrological Year	Scenario	Number of days water table is above threshold depth					
		PET _{temp}			PET _{trws}		
		0.01 m	0.10 m	0.20 m	0.01 m	0.10 m	0.20 m
1997/8	Calibrated	67	119	143	67	119	143
	L	37	66	86	1	11	35
	ML	24	60	74	0	0	29
	MH	16	42	56	0	0	8
	H	23	57	71	0	0	0
1998/9	Calibrated	88	170	175	88	170	175
	L	78	137	149	1	6	7
	ML	74	126	130	0	0	0
	MH	72	120	129	0	0	0
	H	37	80	88	0	0	0

Table 3. Number of days when simulated water tables are above threshold depths close to the ground surface during each complete hydrological year of the simulation period for each scenario

Scenario	1997/8			1998/9		
	Minimum (m) [†]	Maximum (m)	Range (m) [‡]	Minimum (m) [†]	Maximum (m)	Range (m) [‡]
Calibrated	0.00 (-0.20)	0.96	0.96 (1.16)	0.00 (-0.16)	0.99	0.99 (1.15)
L _{temp}	0.00 (-0.04)	0.95	0.95 (0.99)	0.00 (-0.12)	1.01	1.01 (1.13)
ML _{temp}	0.00 (-0.04)	0.98	0.98 (1.02)	0.00 (-0.11)	1.04	1.04 (1.15)
MH _{temp}	0.00	1.03	1.03	0.00 (-0.11)	1.03	1.03 (1.14)
H _{temp}	0.00 (-0.05)	1.00	1.00 (1.05)	0.00 (-0.04)	1.07	1.07 (1.11)
L _{trws}	0.01	1.03	1.02	0.01	1.11	1.10
ML _{trws}	0.13	1.03	0.90	0.21	1.19	0.98
MH _{trws}	0.18	1.05	0.87	0.31	1.26	0.95
H _{trws}	0.25	1.07	0.82	0.48	1.22	0.74

[†] Negative values in brackets indicate the depth of surface water, [‡] Numbers in brackets are the range when surface water depth is included as the maximum water table elevation.

Table 4. Minimum, maximum and range of groundwater depths for each complete hydrological year of the simulation period for each scenario (see text for definition of hydrological years)

Hydrological characteristic	Climate Change Scenarios	
	PET _{temp}	PET _{trws}
Water table	The autumn/winter rise is delayed and the spring recession begins earlier so duration of high water tables is reduced.	The autumn/winter rise is delayed further and water tables fall even earlier in spring.
	Winter water tables still intercept the ground surface.	Winter peak levels are lower and rarely intercept the ground surface (-0.01 m, -0.20 m, -0.48 m) [†] .
	Summer water tables are generally lower (0.01 m, -0.04 m, -0.08 m) [†] .	Summer water tables are consistently lower for progressively higher emissions scenarios (-0.07 m, -0.15 m, -0.27 m) [†] .
	The range of depths increases although there is limited variation between scenarios.	The ranges of depths are generally lower and decline through the progressively higher emissions scenarios.
Ditch water	The autumn/winter rise is delayed whilst the spring recession begins earlier so that the duration of high water levels is reduced.	The autumn/winter rise is delayed further whilst the spring recession begins earlier. The duration of high ditch water levels is reduced.
	Peak winter water levels in dry years (1997/8) fail to reach the elevation of the sluices. In wetter years (1998/9) peak water levels exceed the elevation of the sluices and some small winter peaks associated with individual rain events are higher.	Peak winter levels are lower (-0.19 m, -0.21 m, -0.24 m) [†] and never exceed the elevation of the sluices. There is little variation between scenarios.
	Summer ditch water levels are generally lower and decline through each of the progressively higher emissions scenarios (0.01 m, -0.06 m, -0.21 m) [†] .	Summer ditch water levels are lower and decline through each of the progressively higher emissions scenarios (-0.24 m, -0.30 m, -0.39 m) [†] .
	The range in ditch water levels increases slightly but there is little variation between scenarios.	The range in ditch water levels increases through each of the progressively higher emissions scenarios.
Surface inundation	The winter expansion of inundation of all depths is delayed whilst the subsequent recession begins earlier so that the duration of inundation is reduced. Peak extents of inundation of all depths are reduced. Only in wet years are modest areas flooded deeply (0.3 m and 0.4 m) for short periods. Otherwise deeper inundation is limited to low-lying areas adjacent to ditches.	Extensive areas of winter inundation are virtually eliminated with surface water being restricted to the lowest areas adjacent to ditches.
	Some areas retain shallow surface water throughout spring and summer although their extent is reduced.	The maintenance of shallow surface water in some areas throughout spring and summer is eliminated.

[†] Minimum, mean and maximum change from calibrated model results derived for the two complete hydrological years of the simulation period.

Table 5. Summary of hydrological changes within the Elmley Marshes associated with the simulation of the climate change scenarios using PET_{temp} and PET_{trws}

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Figure 4. Simulated groundwater depths within the Elmley Marshes for the calibrated model and UKCIP02 Climate Change Scenarios for 2050: (a) using PET_{temp} , (b) using PET_{trws} . Observed groundwater depths used in model calibration are also shown ($R^2 = 0.80$)

Figure 5. Depth–duration graphs for simulated groundwater depths within the Elmley Marshes for the calibrated model and UKCIP02 Climate Change Scenarios for 2050: (a) using PET_{temp} , (b) using PET_{trws} .

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Figure 7. Water level–duration graphs for simulated ditch water levels within the Elmley Marshes for the calibrated model and UKCIP02 Climate Change Scenarios for 2050: (a) using PET_{temp} , (b) using PET_{trws} .

Figure 8. Extent of flooding for a range of maximum water depths for the calibrated model results and each UKCIP02 Climate Change Scenario using PET_{temp} : (a) 0.1 m depth, (b) 0.2 m depth, (c) 0.3 m depth, (d) 0.4 m depth.

Figure 9. Extent of flooding for a range of maximum water depths for the calibrated model results and each UKCIP02 Climate Change Scenario using PET_{trws} : (a) 0.1 m depth, (b) 0.2 m depth, (c) 0.3 m depth, (d) 0.4 m depth.

Figure 10. Impacts of climate change on flooding during the breeding season of wading birds: (a) mean extent of flooding of 0.1 m maximum depth in April and May - calibrated model and PET_{temp} scenarios, (b) mean extent of flooding of 0.1 m maximum depth in April and May - calibrated model and PET_{trws} scenarios, (c) simulated flood extent on 01/05/1999 - calibrated model, (d) simulated flood extent on 01/05/1999 - L_{temp} scenario, (e) simulated flood extent on 01/05/1999 - H_{temp} scenario, (f) simulated flood extent on 01/05/1999 - L_{trws} scenario, (g) simulated flood extent on 01/05/1999 - L_{trws} scenario.

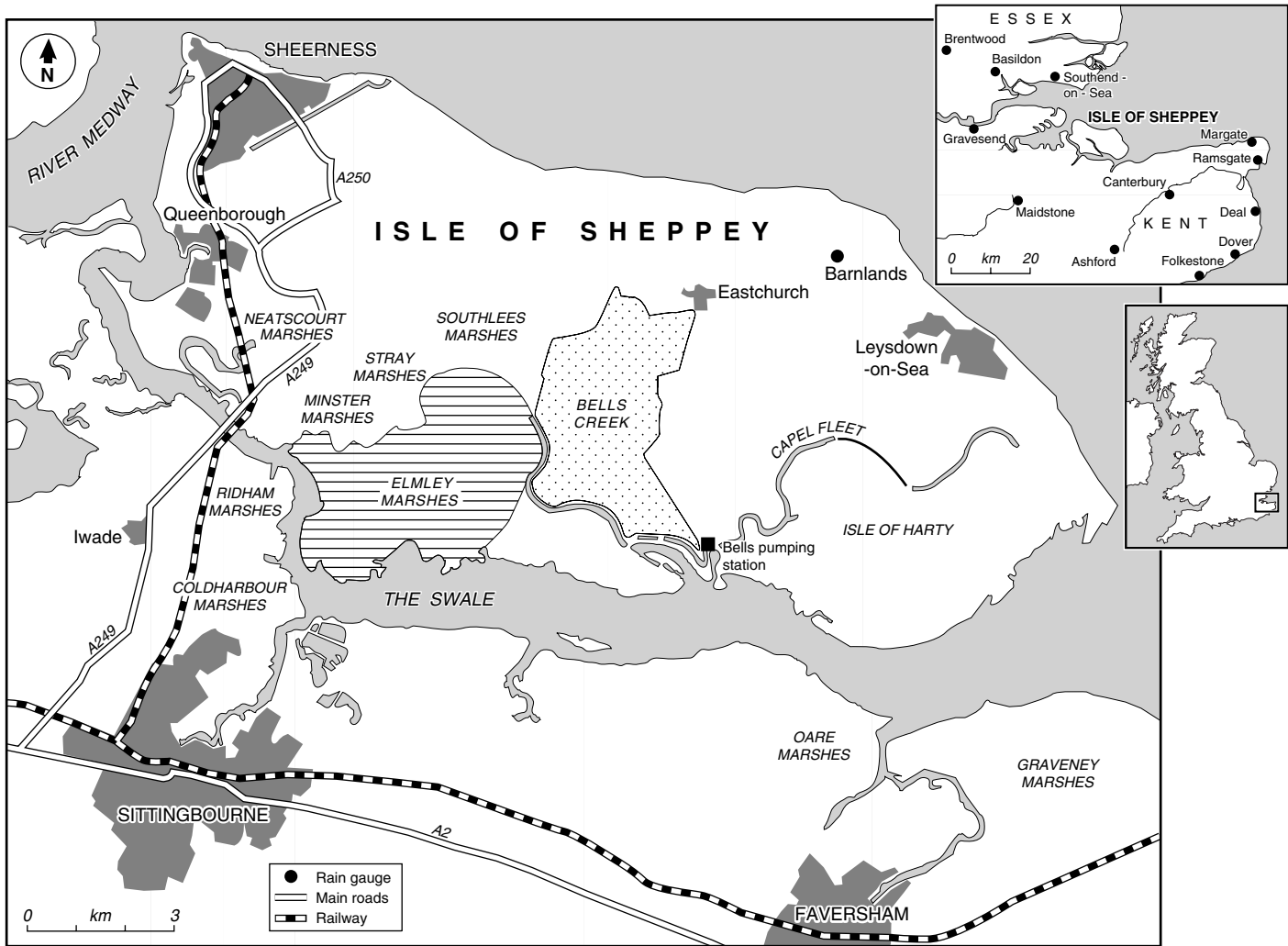
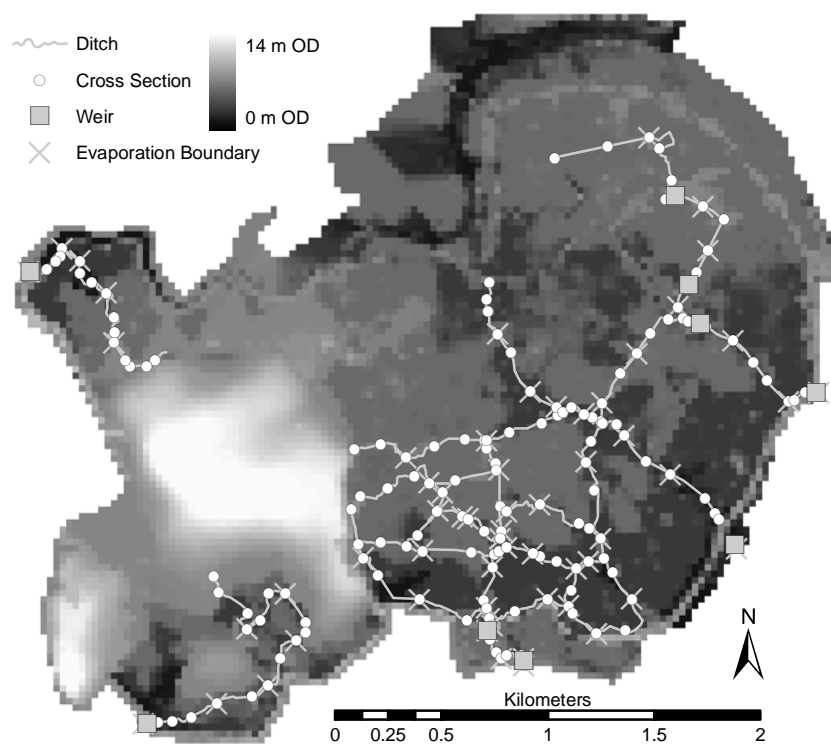


Figure 2. The 30 m \times 30 m MIKE SHE topographic grid and MIKE 11 model of the Elmley Marshes



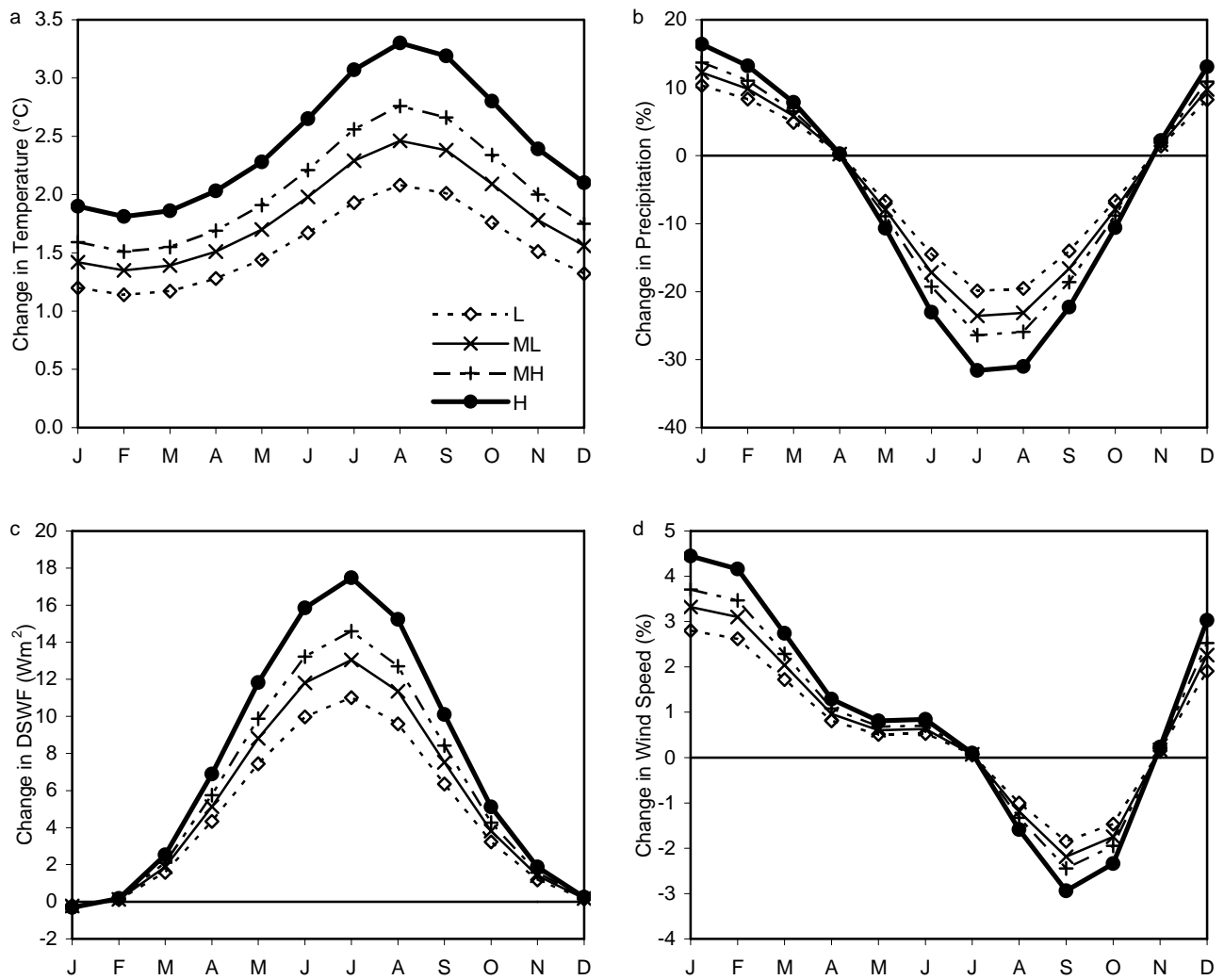


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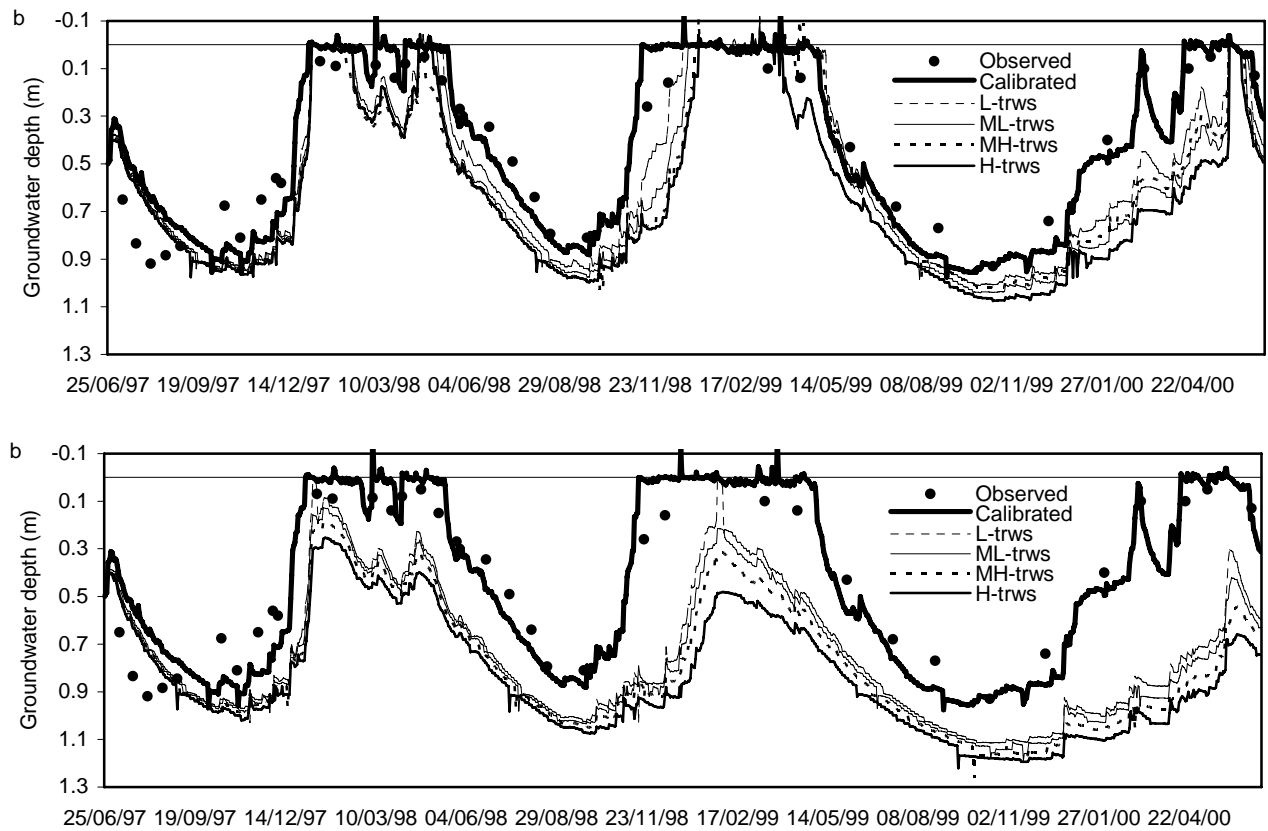


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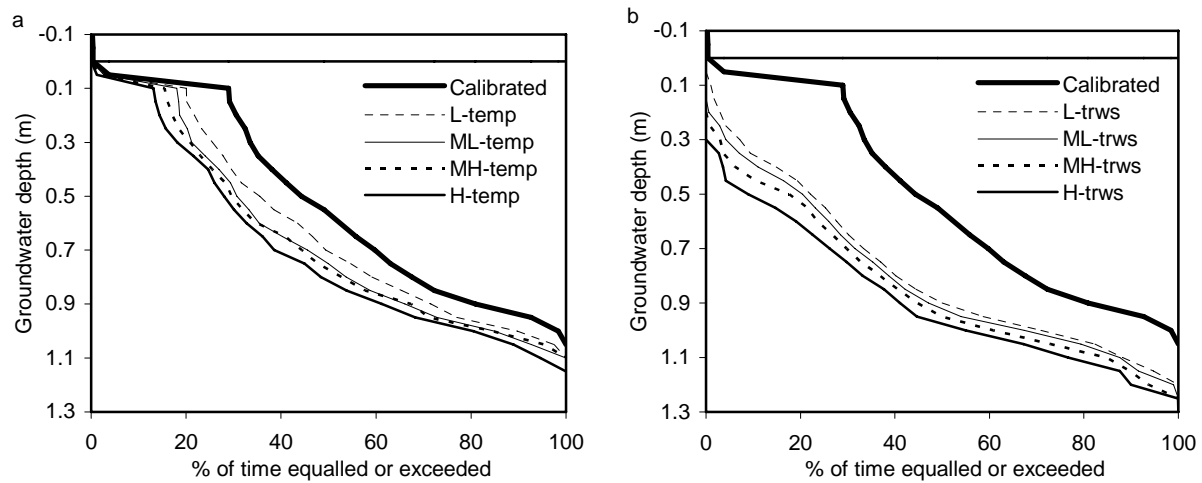


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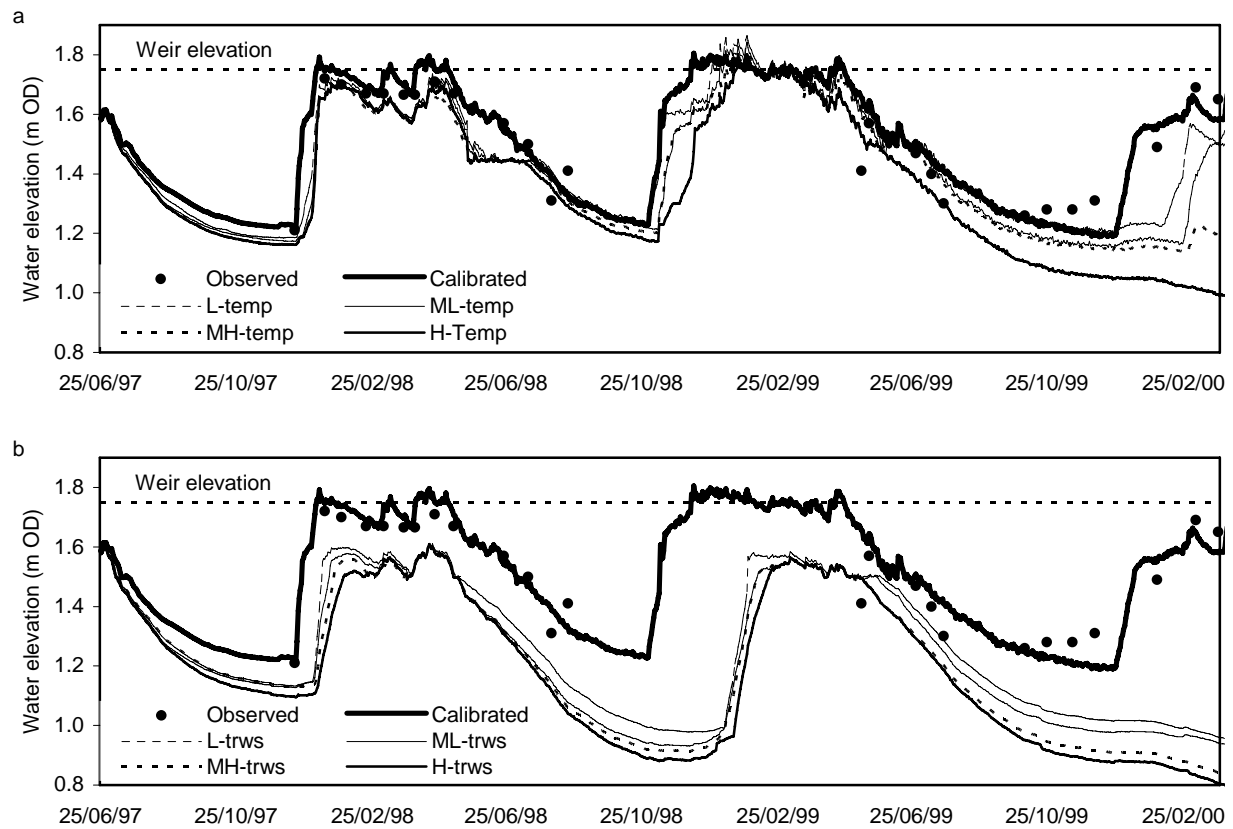


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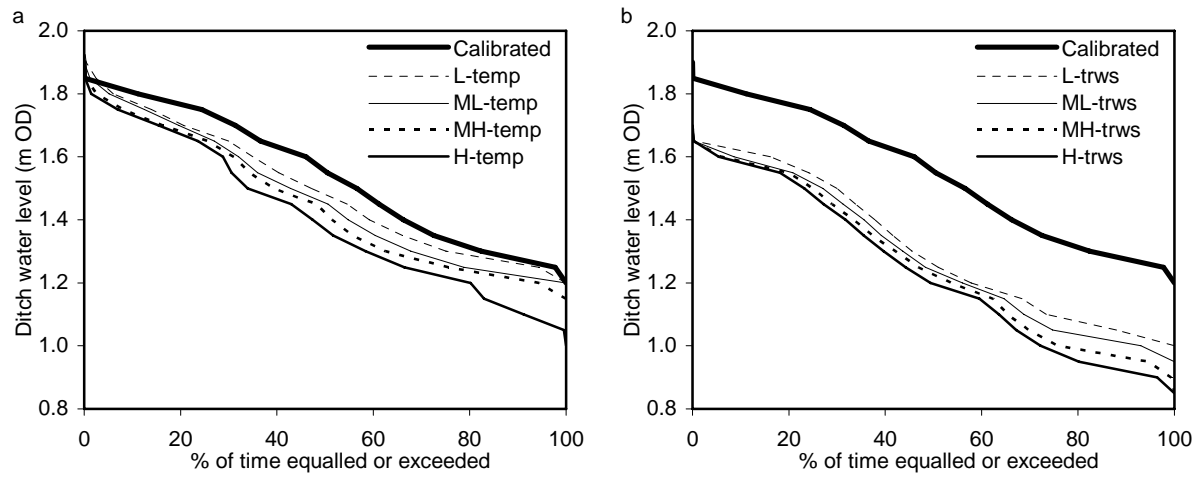


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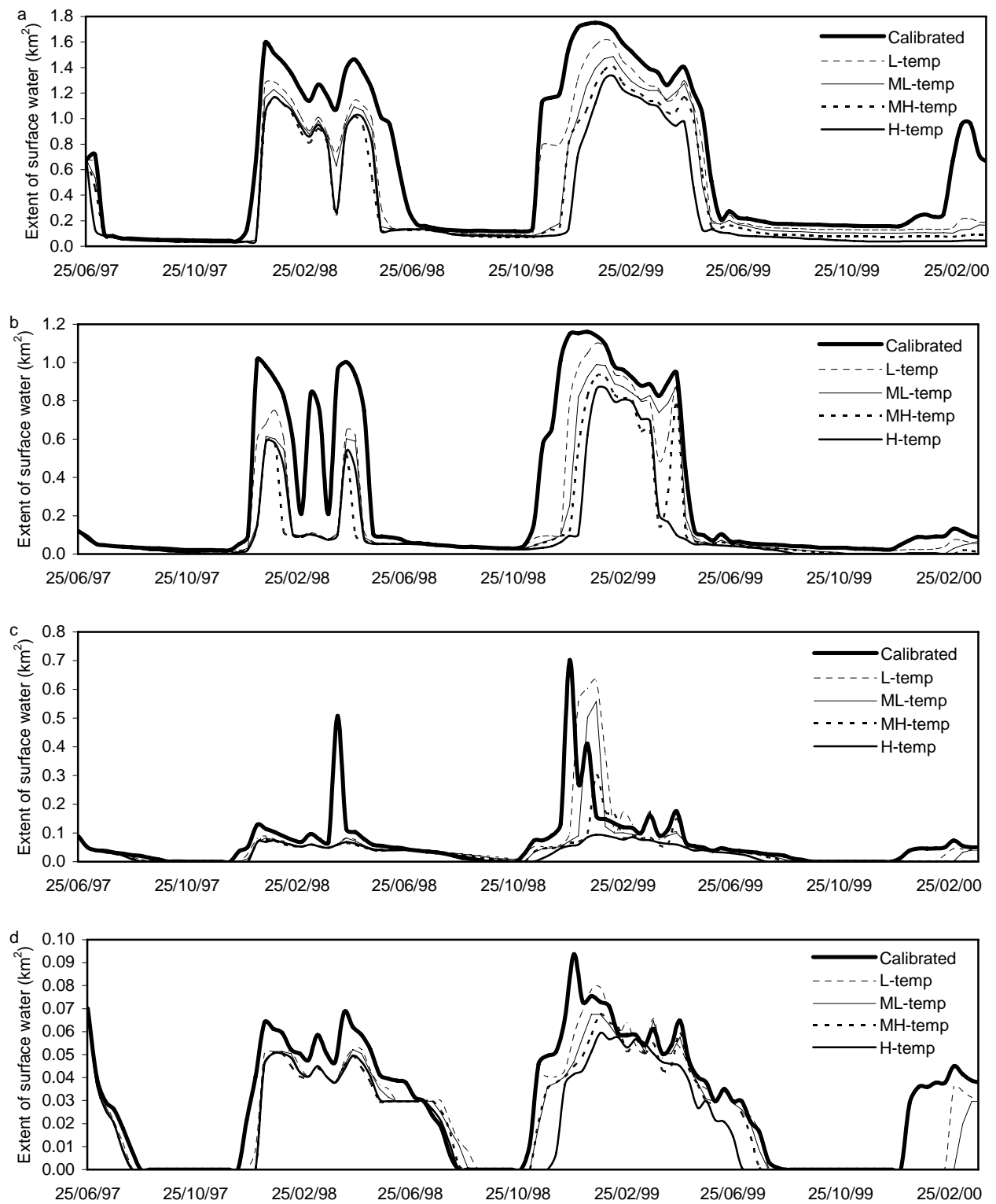


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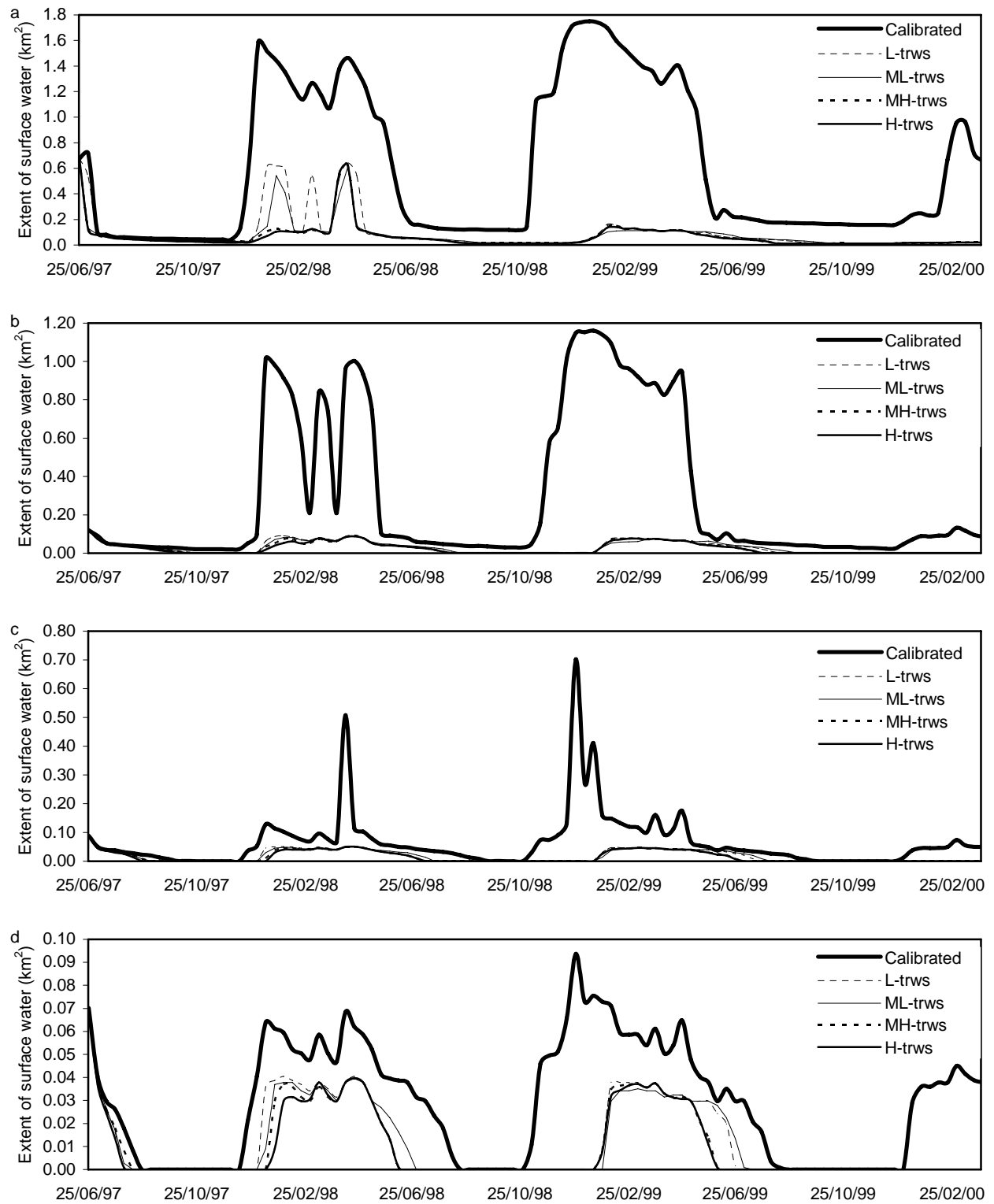


Figure 9. Extent of flooding for a range of maximum water depths for the calibrated model results and each UKCIP02 Climate Change Scenario using PET_{trws} : (a) 0.1 m depth, (b) 0.2 m depth, (c) 0.3 m depth, (d) 0.4 m depth.

